BOREAL ENVIRONMENT RESEARCH 16: 158–166
ISSN 1239-6095 (print) ISSN 1797-2469 (online)

Helsinki 29 April 2011

Soil-to-plant transfer of uranium and its distribution between plant parts in four boreal forest species

Päivi Roivainen*, Sari Makkonen, Toini Holopainen and Jukka Juutilainen

University of Eastern Finland, Department of Environmental Science, P.O. Box 1627, FI-70211 Kuopio, Finland (corresponding author's e-mail: paivi.roivainen@uef.fi)

Received 8 Feb. 2010, accepted 30 Aug. 2010 (Editor in charge of this article: Jaana Bäck)

Roivainen, P., Makkonen, S., Holopainen, T. & Juutilainen, J. 2011: Soil-to-plant transfer of uranium and its distribution between plant parts in four boreal forest species. *Boreal Env. Res.* 16: 158–166.

Uranium (U) can be released to the environment through the entire nuclear fuel cycle. U uptake by plants is an important process for possible adverse effects in ecosystems. The soil-to-plant transfer of natural U and its distribution across plant parts were investigated in May lily (*Maianthemum bifolium*), narrow buckler fern (*Dryopteris carthusiana*), rowan (*Sorbus aucuparia*) and Norway spruce (*Picea abies*). Concentration ratios (CR) between plant and soil were calculated. The CRs for roots were higher than those for the above-ground parts of the plants. Soil pH was the only soil parameter showing an effect on CRs. No significant differences were noticed between species. The CRs observed were consistent with those reported previously in other forest types. The pooled values of 0.06 for roots and 0.005 for stems/petioles and leaves/needles can be considered as good estimates of CR values to be used in modelling the U uptake in boreal forest species.

Introduction

Nuclear power provides about 15% of the world's electricity. The current fission-based nuclear fuel cycle requires uranium (U), and therefore possibilities for opening new mines for U fuel production are continuously sought. U can be released to the environment through the entire nuclear fuel cycle from mines to spent nuclear fuel disposal. The uptake of U by plants is a process that could lead to adverse effects in ecosystems. However, this process is not sufficiently understood at present.

Radioecological modelling is used for predicting long-term behaviour of radionuclides in the environment. Most of the existing models are intended for simulating deposition of radionuclides from the atmosphere, but recently models have been developed also for scenarios of belowground contamination (Avila 2006). The existing models are generally compartment models such as the forest model of Avila (2006) which includes the compartments of soil, litter, tree wood, tree leaves, understory and fauna. Uptake into plants is commonly described by a concentration ratio (CR), often called a transfer factor, which is calculated by dividing plant U concentration by soil U concentration (Avila 2006, IAEA 2010).

Soil pH is one of the most important factors affecting the behaviour of U in soils (Ebbs *et al.* 1998, Echevarria *et al.* 2001, Koch-Steindl and Pröhl 2001). Clay content and organic matter content had no significant effects on the sorp-

tion of U in a study by Echevarria *et al*. (2001). Vandenhove *et al*. (2007) found no single soil parameter to correlate with U uptake by ryegrass (*Lolium perenne* cv. Melvina).

UO_2^{2+} is the U species most readily taken up and translocated by plants (Ebbs *et al*. 1998). U accumulates in the roots of plants (Shahandeh and Hossner 2002, Shtangeeva 2010), which results in different U concentrations in different plant parts and thus leads to differences in CRs between plant parts. Variation in U uptake and translocation between different plant groups has also been reported. According to Shahandeh and Hossner (2002), dicotyledonous plant species tend to accumulate more U than monocotyledonous species.

The transfer of U from soil to plant has usually been studied with agricultural plants (e.g. Shahandeh and Hossner 2002, Duquene *et al*. 2006). CRs from such studies may have limited validity for describing the behaviour of U in boreal forests, and only few data are available from boreal forest plant species in natural conditions (e.g. Morton *et al*. 2002).

Here, U uptake by four different boreal plant species was studied at a U occurrence, which provides a natural laboratory for conducting uptake studies in a boreal forest setting. We investigated the soil-to-plant transfer of natural U and its distribution across plant parts in four species. As the existing knowledge of the transfer of U into forests plants is scarce, the goal of our study was to collect samples representing different plant types (understory, deciduous trees and coniferous trees). The most abundant species of each group were selected in order to get enough sample material. The selected plants are also common in boreal forest. May lily (*Maianthemum bifolium*) is a monocotyledonous herb, narrow buckler fern (*Dryopteris carthusiana*) a fern, rowan (*Sorbus aucuparia*) a dicotyledonous deciduous tree and Norway spruce (*Picea abies*) a coniferous tree. Our objectives were to: (i) determine the distribution of U across different plant parts, (ii) determine CRs for boreal forest plant species and their dependence on soil properties, (iii) investigate possible differences between plant species in their U uptake properties.

Material and methods

Site description

The sampling site was a U occurrence located in Nilsiä, eastern Finland (63°04´N, 27°54´E). The site was a small (circa 0.04 km^2) herb-rich forest (classified as *Oxalis–Maianthemum* site type) entirely surrounded by agricultural land. During the 1960s small-scale ore prospecting was carried out in the area. A 100-m-long excavation pit with a small pond at one end is present in the area as a result of the prospecting. There are also uranium-rich rocks at the soil surface around the pit.

The dominating tree species in the study area are Norway spruce (*Picea abies*), rowan (*Sorbus aucuparia*) and common aspen (*Populus tremula*). Understory species include common wood sorrel (*Oxalis acetosella*), May lily (*Maianthemum bifolium*), narrow buckler fern (*Dryopteris carthusiana*) and oak fern (*Gymnocarpium dryopteris*).

Soil samples

The soil samples were collected from 29 systematically selected sampling points in June 2007. A grid consisting of ten squares was established to assist the sampling point selection. Six squares (size 40 m × 40 m) were around the pit and four squares (size 60 m × 60 m) were in the untouched area. Three sampling points in a triangle-shaped arrangement were systematically selected inside each square, except only two points were selected from the square furthest away from the pit because it was partially outside the forest. Thus 18 sampling points were within 30 m from the pit and 11 sampling points were located at a 40–100 m distance from the pit. There were a lot of rocks on the soil surface around the pit but the disturbance caused by ore prospecting was otherwise no more distinguishable. One sample of litter and topsoil was collected from each sampling point. The litter samples consisted of leaves and needles which were slightly decomposed but still identifiable. The litter samples (fresh weight 50–100 g) were oven dried (60 °C) for at least 24 hours. The dried

samples were milled and dry matter content was analysed (24 h at 105 °C).

The distinction between organic and mineral soils at the site was not clear. The topsoil was collected with a spade to a depth of 100 mm within an area of 100 mm × 100 mm. This was assumed to be the rooting depth of the understory species. Fresh weight of the collected soil samples varied from 200–600 g depending on the amount of rocks. The topsoil samples were oven dried (40 °C) for seven days. After drying the samples were sieved to diameter fractions < 2 mm and > 2 mm. The soil fraction < 2 mm was used for analyses. Dry matter content (24 h at 105 °C), pH (soil:water v:v 1:5) and organic matter (OM) content (3 h at 550 °C) were analysed. Particle size distribution was analysed using the pipette method according to ICP Forests (2006) with slight modifications. Organic matter was oxidised by hydrogen peroxide H_2O_2 and particles were dispersed by sodium hexametaphosphate $(NaPO_3)_6$. The sand fraction was removed by wet sieving (62.5 μm), after which silt and clay fractions were measured by sedimentation. Two replicates of each sample were used in the analysis of soil OM content and particle size distribution.

Plant samples

Plant samples were collected in June 2007. The collected plant species were May lily (*Maianthemum bifolium*), narrow buckler fern (*Dryopteris carthusiana*), rowan (*Sorbus aucuparia*) and Norway spruce (*Picea abies*). The plants growing closest to the selected soil sampling point were sampled. Each species was collected if a sufficient amount of plant material occurred within five metres from the selected sampling point. May lily was collected from 19 points, narrow buckler fern from 27 points, rowan from 28 points and Norway spruce from 26 points. In addition to roots (diameter fractions of < 2 mm and > 2 mm) leaves of rowan and needles of Norway spruce were collected. The rowans growing at the site were mostly saplings and therefore the root and leaf samples were collected from two to three saplings per point. Leaves were collected without branches. The Norway spruces were mature trees and each sample was collected from one individual. The second and third year needles were collected with the twigs from the lower canopy. The branches were removed after drying the samples. Samples of May lily and narrow buckler fern were divided into three parts (root, stem/petiole, and leaf) in the field. The root sample of narrow buckler fern consisted of both the rhizome and fine roots. The fresh weight of each sample was 50 g. All of the root samples were washed with milliQ-water before drying. Vegetation samples were oven-dried (60 °C) for at least 24 hours. After drying the samples were milled and dry matter content was analysed (24 h at 105 °C).

Chemical analysis

Inductively coupled plasma mass spectroscopy (ICP-MS) (Perkin Elmer Sciex Elan 6000) measurements were conducted in the laboratory of Labtium Ltd. in Espoo, Finland, providing pseudototal concentrations of U after nitric acid digestion (EPA 3051) in a microwave oven. An estimate of the mobile fraction of U in the soil samples was obtained by ICP-MS (Perkin Elmer Sciex Elan 5000) analyses after 1 M ammonium acetate (NH_4Ac, buffered at pH 4.5) leach (Räisänen *et al*. 1997). The detection limit for uranium in both cases was 0.01 mg kg^{-1}.

Data analysis

All the results were corrected relative to the dry matter content. If the U concentration in a plant part was below the detection limit, the value corresponding to half the detection limit (0.005 mg kg^{-1}) was used in calculations. CRs for U were calculated separately for root, stem/petiole and leaf/needle and for each species as follows:

$$CR_{t,p} = [U]_p [U]^{-1}_{total} \quad (1)$$

where $[U]_p$ is the concentration (mg kg^{-1} (DW)) of U in plant part p (p = root, fine root, stem, petiole, leaf or needle) and $[U]_{total}$ is the total concentration (mg kg^{-1} (DW)) of U in soil.

$$CR_{m,p} = [U]_p [U]^{-1}_{mobile} \quad (2)$$

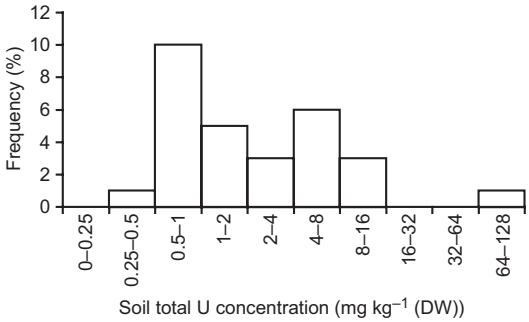

Fig. 1. The distribution of soil total U concentration.

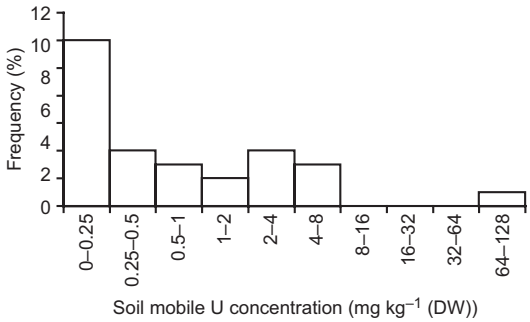

Fig. 2. The distribution of soil mobile U concentration.

where $[U]_{mobile}$ is the mobile concentration (mg kg^{-1} (DW)) of U in soil.

SPSS 14.0 for Windows was used for statistical analysis. A Kruskall-Wallis test followed by a Mann-Whitney *U*-test with the Bonferroni correction as a post hoc test was used for group comparisons, and Spearman's ρ was calculated for correlation matrices. A multiple regression analysis was carried out to investigate the dependence of mobile soil U concentration on other soil properties. The data was log-transformed for the multiple regression analysis in order to have normally distributed data. The effects of soil properties, plant species and plant part on CRs were tested with a general linear model and regression coefficients (*B* values) were calculated. Results or differences were considered statistically significant when $p \leq 0.05$.

Table 1. Medians and ranges of dry/fresh weight ratios in soil, litter and plant samples.

	Median	Range
Topsoil (n = 29)	0.83	0.65–0.93
Litter (n = 29)	0.46	0.31–0.81
May lily root (n = 19)	0.20	0.16–0.22
May lily stem (n = 19)	0.14	0.12–0.16
May lily leaf (n = 19)	0.18	0.16–0.22
Fern root (n = 27)	0.19	0.16–0.26
Fern petiole (n = 27)	0.15	0.13–0.17
Fern leaf (n = 27)	0.23	0.17–0.30
Rowan coarse root (n = 28)	0.36	0.30–0.44
Rowan fine root (n = 28)	0.33	0.26–0.51
Rowan leaf (n = 28)	0.25	0.21–0.31
Norway spruce coarse root (n = 26)	0.37	0.25–0.58
Norway spruce fine root (n = 26)	0.37	0.28–0.48
Norway spruce needle (n = 26)	0.46	0.37–0.53

Geometric means (GM) and geometric standard deviations (GSD) were used here to describe the distributions, with exception of soil properties for which arithmetic means were calculated.

Results

Dry matter content of the samples

All CR values in this paper are expressed per dry weight. However, the CR values are sometimes provided per wet weight. To allow comparisons with CR values based on wet weights, the dry matter content of the soil and plant samples is given in Table 1.

Uranium in soil

There were great variations in both total and mobile U concentrations in soil (Figs. 1–2), and they were significantly correlated with each other (Spearman ρ = 0.898, $p < 0.001$, n = 29). The highest U concentration was observed 20 m away from the pit and near a trail through which the rocks were transported away from the site during the ore prospecting. The next highest U concentrations were measured close to the pit. However, some of the lowest concentrations were also measured close to the pit and there were also high concentrations in the area where the ground was considered untouched. U concentration in litter varied from 0.32 to 34.70 mg kg^{-1} (DW) and was significantly correlated with mobile and total U concentrations in soil: Spearman ρ's of 0.584 (p

Table 2. Arithmetic mean values, medians and ranges of soil properties at the study site (n = 29).

Soil property	Mean	Median	Range
pH	4.4	4.4	4.0–5.1
Organic matter (%)	13.1	11.3	2.7–36.9
Clay (%)	9.6	9.4	3.6–18.3
Silt (%)	9.8	8.5	1.9–33.7

= 0.001, n = 29) and 0.561 (p = 0.002, n = 29) respectively. The ratio between litter U and total U concentrations in soil ranged from 0.007 to 2.01 (GM 0.12, GSD 3.05) and the ratio between litter U and mobile U concentrations in soil varied from 0.01 to 7.46 (0.45, 4.04).

OM content, pH, clay content and silt content were the measured soil parameters (Table 2). None of these parameters showed significant correlation with the total or mobile soil U concentration (data not shown). A multiple regression analysis showed that soil total U concentration (p < 0.001) was the only variable to significantly explain soil mobile U concentration. In the first analysis with all data included (n = 29), OM content (p = 0.007) and clay content (p < 0.001) were also significantly associated with soil mobile U concentration, which increased with increasing OM content and decreased with increasing clay content. However, this finding was strongly affected by one plot with a high U concentration in soil. When the data from this one plot were removed from the analysis, the effects of OM and clay content were no longer statistically significant and the direction of the effect of OM changed sign.

Uranium in plants

In all the studied plant species, the U concentration in roots was significantly higher than that in leaves/needles (p < 0.05, df = 2 for all species) (Table 3). In Norway spruce and rowan, the U concentration in fine roots was significantly higher than in coarse roots (p < 0.05). In May lily and narrow buckler fern, the U concentration appeared to be similar in stem/petiole and leaves (Table 3). Root-to-leaf ratios of May lily were significantly lower than those of fern (p = 0.001) and rowan (p < 0.001), when the ratio in trees was calculated based on the U concentrations of coarse roots (Table 3). The fine root-to-leaf ratios in Norway spruce (p < 0.001) and rowan (p = 0.005) were higher than coarse root-to-leaf ratios.

Plant U concentration showed a better correlation with mobile than total soil U. Spearman correlation coefficients between soil mobile U concentration and plant U concentration were, for all plant parts, higher than the correlation coefficients between total soil U concentration and plant U concentration (Table 4). This difference was statistically significant (paired t-test: p < 0.05, n = 12, df = 11).

Plant *vs.* substrate concentrations (concentration ratios)

In all the studied plant species, the CR values for roots were higher than the corresponding values for stems/petioles and leaves/needles (p < 0.05, df = 2) (Table 5). The CR values were also higher for fine roots of trees than the corresponding CR values for coarser roots (p < 0.05). The CR values

Table 3. Geometric means (and geometric standard deviations) of U concentrations (mg kg^{-1} (DW)) in root, stem/petiole and leaf/needle and root-to-leaf ratios in different plant species. In case of trees, "root" refers to coarse root.

	May lily (n = 19)	Fern (n = 27)	Rowan (n = 28)	Norway spruce (n = 26)
Root	0.09 (4.13)	0.17 (4.79)	0.15 (4.50)	0.11 (4.14)
Fine root			0.51 (5.08)	0.62 (5.88)
Stem/petiole	0.01 (2.20)	0.01 (1.88)		
Leaf/needle	0.02 (3.68)	0.01 (2.02)	0.01 (1.35)	0.01 (2.63)
Root-to-leaf ratio	5.1 (2.28)	18.7 (3.49)	24.9 (4.29)	9.5 (4.29)
Fine root-to-leaf ratio			86.5 (5.01)	54.6 (4.61)

Table 4. Spearman correlations (ρ) between U concentration in different plant parts and total and mobile U concentration in soil. Values set in boldface are statistically significant at $p \leq 0.05$.

	$[U]_{total}$ in soil	$[U]_{mobile}$ in soil
May lily root ($n = 19$)	**0.725**	**0.792**
May lily stem ($n = 19$)	**0.497**	**0.599**
May lily leaf ($n = 19$)	**0.573**	**0.679**
Fern root ($n = 27$)	**0.500**	**0.593**
Fern petiole ($n = 27$)	0.333	**0.457**
Fern leaf ($n = 27$)	**0.384**	**0.454**
Rowan coarse root ($n = 28$)	**0.480**	**0.500**
Rowan fine root ($n = 28$)	**0.625**	**0.648**
Rowan leaf ($n = 28$)	0.022	0.194
Norway spruce coarse root ($n = 26$)	**0.418**	**0.564**
Norway spruce fine root ($n = 26$)	0.373	**0.500**
Norway spruce needle ($n = 26$)	0.187	0.265

for stems/petioles in May lily and narrow buckler fern were similar to the CR values for leaves.

The CR values of plant parts were of the same magnitude in all plant species studied. The general linear model analysis used to investigate dependences of CR on plant species, plant part and soil properties, identified plant part as a significant explanatory variable ($p = 0.001$ for CR based on mobile U and $p < 0.001$ for CR based on total U), but plant species did not affect CR ($p = 0.295$ for CR_m and $p = 0.286$ for CR_t). CR also decreased significantly with increasing pH ($p = 0.022$ for CR_m and $p = 0.019$ for CR_t), but no relation were observed with OM, clay or silt content (Table 6).

A possible contamination of plant samples with soil was evaluated using plant titanium (Ti) concentration as an indicator of that contamination, as plants take up only small amounts of Ti (Nisbet and Shaw 1994, Cook *et al*. 2007). Con-

Table 5. Geometric mean (and geometric standard deviation) of concentration ratios for different plant species and for pooled values of all species. The CRs are given for both total and mobile concentration of U in soil. In case of trees, "root" refers to coarse roots.

Type of CR	May lily $n = 19$	Fern $n = 27$	Rowan $n = 28$	Norway spruce $n = 26$	Pooled
Soil-to-root					
Soil total U	0.04 (2.36)	0.08 (3.86)	0.07 (3.77)	0.06 (3.88)	0.06 (3.63)
Soil mobile U	0.18 (3.22)	0.29 (4.50)	0.26 (4.68)	0.22 (4.29)	0.24 (4.25)
Soil-to-fine root					
Soil total U			0.24 (3.47)	0.32 (5.23)	0.27 (4.27)
Soil mobile U			0.91 (3.94)	1.25 (5.47)	1.06 (4.63)
Soil-to-stem/petiole					
Soil total U	0.004 (2.99)	0.004 (3.18)			0.004 (3.16)
Soil mobile U	0.02 (5.06)	0.01 (4.89)			0.01 (4.90)
Soil-to-leaf/needle					
Soil total U	0.008 (2.71)	0.004 (3.15)	0.003 (3.54)	0.006 (3.68)	0.005 (3.51)
Soil mobile U	0.03 (3.77)	0.02 (4.88)	0.01 (5.74)	0.02 (5.19)	0.02 (5.13)

Table 6. The dependence of concentration ratios on soil properties in general linear models based on soil total U or soil mobile U. *B* values (regression coefficients) of different model parameters with their SEs and *p*'s are shown.

	Model based on total U			Model based on mobile U		
	B	SE	*p*	*B*	SE	*p*
Intercept	5.621	1.915	0.004	29.548	10.453	0.005
pH	–0.993	0.422	0.019	–5.306	2.305	0.022
Organic matter (%)	–0.340	1.841	0.854	–2.405	10.046	0.811
Clay (%)	–9.565	5.321	0.073	–47.732	29.041	0.101
Silt (%)	3.205	2.880	0.267	14.570	15.721	0.355

tamination of leaves/needles and stems/petioles by soil was generally low. Berrow (1988) suggested that plants with Ti concentration higher than 10 mg kg^{-1} (DW) should be considered soil-contaminated. This value was exceeded in a small proportion of the above-ground plant samples. Also, Ti concentrations in these plant parts were usually 0.1%–1.6% of the Ti concentration in soil. In May lily, however, higher contamination by soil was found in two leaf samples (8.3% and 11.0%) and in three stem samples (1.9%, 2.8% and 3.2%). However, removing these samples from the data did not change the CRs reported in Table 5. In root samples, Ti concentration generally exceeded 10 mg kg^{-1} (DW), and root Ti concentrations as a percentage of soil Ti concentrations were particularly high in narrow buckler fern roots (1.5%–29.8%), in the fine roots of rowan (1.2%–34.7%), and in the fine roots of Norway spruce (1.1%–49.5%). Because of the apparent contamination of root samples by soil, the CRs for roots must be interpreted to reflect root total U, which consists of U taken up into root tissue and of U in soil particles that persist on root surface even after washing.

Discussion

The total U concentrations at the study site were mostly within the average range of U concentrations measured in different types of soil (1–10 mg kg^{-1} Koch-Steindl and Pröhl 2001). Thus, the results are valid for transfer of U at background uranium levels.

The measured U concentrations in stems/petioles and leaves/needles were generally low and some of them even below the detection limit of the chemical analysis method used. Including the results under the detection limit to calculations did not have a significant effect on the CRs. The pooled CR values were 0.005 for soil-to-stem/petiole and 0.006 for soil-to-leaf/needle when concentrations under the detection limit were excluded from the analysis. These values were not remarkably different as compared with those presented in Table 5 when also the high GSD was taken into account.

In our study, root-to-leaf ratios varied from 5 in May lily to 25 in rowan. Shahandeh and Hossner (2002) reported root U concentrations 30 to 50 times higher than the shoot U concentrations in several agricultural plants. In our study, the root-to-leaf ratio of May lily was significantly lower than the corresponding ratios of fern and rowan. The low root-to-leaf ratio in May lily is mainly explained by low U concentration in its roots; concentration of U in its stems and leaves was at the same level or slightly higher than in other species. May lily was the only monocotyledonous species used in this study. Monocotyledonous plants have been reported to take up less uranium than dicotyledonous species in a study of agricultural plants (Shahandeh and Hossner 2002). Our data show that the low uptake in May lily was limited to roots only, while translocation to the shoots was higher than in the other species.

The CRs were clearly different for different plant parts. The CRs for roots were higher than the corresponding values for stems/petioles and leaves/needles which is in agreement with Sheppard and Evenden (1988), Shahandeh and Hossner (2002) and Shtangeeva (2010). The CRs for roots were 5 to 20 times higher than those for stems/petioles and leaves/needles. It was also observed that CRs for fine roots were 3.5 to 6-fold higher than those for the coarser roots. Thiry *et al*. (2005) found a similar pattern in 35-year old Scots pine trees growing on a revegetated U-mining heap in Germany. In their study the U content in bulk root samples was about 1.5 times higher for fine roots (diameter < 2 mm) than for coarser roots (diameter 2–10 mm).

In our study, plant species did not significantly affect the CRs. Sheppard *et al*. (2006) listed CRs between soil total U and various plant types and found that the GM of CRs was 0.004 (GSD 16.3) for trees, 0.01 (8.7) for native browse, 0.01 (5.8) for forages, 0.007 (5.2) for shrubs, and 0.3 (11) for lichens, moss and heather. Lichens, moss and heather had substantially higher CRs than other plants (probably reflecting the plant structure and the ability of these plant species to retain dust), but given the large GSDs the differences between other plant species may not support using different CR values for them in radioecological models (Sheppard *et al*. 2006). Apart from lichens, moss and heather, there are only limited data avail-

able concerning CRs for boreal forest understory plants. Morton *et al*. (2002) reported a CR of 0.0004 for blueberry (*Vaccinium pallidum*). The CR values observed for May lily (0.008) and narrow buckler fern (0.004) in this study are consistent with the values reported for shrubs by Sheppard *et al*. (2006) and also with the values for trees found in this study and reported by Sheppard *et al*. (2006). As the species studied in the present study represented four quite different plant types, the data do not support the need to use different CR values for different boreal forest plant species (apart from lichens, moss and heather). The pooled values shown in the last column of Table 5 can be considered as good estimates of CR values to be used in modelling the uptake of U in boreal forest species. Using different values for stems/petioles and leaves/needles cannot be defended based on the data, and the value for leaves/needles (0.005) is also good for stems/petioles.

Soil pH was the only parameter that had a significant effect on CRs with higher CRs at low pH. Also Ebbs *et al*. (1998) found that pH influences U uptake. They reported the greatest shoot concentration and accumulation in peas when plants were growing at pH 5.0 where uranyl cations are likely to predominate. The U uptake of peas grown at pH 6.0 was less than 20% of the uptake at pH 5.0 and it was even less (5%) at pH 8.0 (Ebbs *et al*. 1998). An increased amount of clay and OM has also been shown to decrease CRs in other studies (Mortvedt 1994). Vandenhove *et al*. (2007) reported that no single soil parameters correlate with the U uptake of plants. They found that the uptake process is complex and explained by several soil factors. They created a multiple regression model suggesting that CR is enhanced when U in soil solution, exchangeable Ca, soil solution Cl, total inorganic C and pH are high and the soil solution NO_3 and OM are low. Vandenhove *et al*. (2007) however questioned the usability of the complex relationship in predictive modelling of biosphere transfer, given the large amount of data required.

U concentrations in different plant parts showed a higher correlation with soil mobile U concentration than soil total U concentration. However, the general linear model based on mobile U did not explain the concentration ratios better than the model based on total U; the R^2 values of the models were 0.132 for the total U model and 0.119 for the mobile U model. The extraction method used in this study (ammonium acetate leach at pH 4.5) gives just one estimate of bioavailability. Several other extraction methods are used to estimate e.g. exchangeable U, U bound to amorphous phases (Vandenhove *et al*. 2007), MnOx-U, FeOx-U and organic U (Shahandeh and Hossner 2002). Vandenhove *et al*. (2007) reported no significant correlation between U soil-to-plant transfer and U recovered in selective soil extractions. For modelling purposes the use of different extraction procedures increases complexity and therefore the CR values based on the total concentration may be the most useful.

Variation of CR values is also of interest for modelling the behaviour of U in the biosphere. In our study, the GSD of CRs varied from 2.36 to 5.74, which is consistent with variation observed in other environments and species. According to Sheppard *et al*. (2006) the GSD typically varies from three to six.

Conclusions

The results of our study show that different CRs are needed for roots than for leaves/needles and stems/petioles when U uptake into plants is modelled. The CRs for leaves/needles were found to be similar in the four very different species studied (a coniferous tree, a dicotyledonous deciduous tree, a monocotyledonous herb and a fern), which is favourable from the point of modelling the uptake of U by forest plants. We consider the pooled values of 0.06 for roots and 0.005 for stems/petioles and leaves/needles as good estimates of CR values to be used in modelling. However, species such as lichens and mosses that are likely to have different CR values were not included in this study. The CRs observed in this study in a natural boreal forest are consistent with results from other settings.

Acknowledgements: This project was funded by a grant from State Nuclear Waste Management Fund and a grant from Maj and Tor Nessling Foundation. We thank Dr. James Blande for the linguistical revision of the manuscript.

References

Avila R. 2006. *Model of the long-term transfer of radionuclides in forests*. Technical report TR-06-08, Swedish Nuclear Fuel and Management Co., Stockholm.

Berrow M.L. 1988. Sampling of soils and plants for trace element analysis. *Anal. Proc*. 25: 116–118.

Ebbs S., Brady D.J. & Kochian L.V. 1998. Role of uranium speciation in the uptake and translocation of uranium by plants. *J. Exp. Bot*. 49: 1183–1190.

Echevarria G., Sheppard M.I. & Morel J. 2001. Effect of pH on the sorption of uranium in soils. *J. Environ. Radioact*. 53: 257–264.

Cook L.L., Inouye R.S., McGonigle T.P. & White G.J. 2007. The distribution of stable cesium in soils and plants of the eastern Snake River Plain in southern Idaho. *J. Arid Environ*. 69: 40–64.

Duquène L., Vandenhove H., Tack F., Van der Avoort E., Van Hees M. & Wannijn J. 2006. Plant-induced changes in soil chemistry do not explain differences in uranium transfer. *J. Environ. Radioact*. 90: 1–14.

IAEA 2010. *Handbook of parameter values for the prediction of radionuclide transfer in terrestrial and freshwater ecosystems*. Technical reports series 472, International Atomic Energy Agency, Vienna.

ICP Forests 2006. *Manual on methods and criteria for harmonized sampling, assessment, monitoring and analysis of the effects of air pollution on forests. Part IIIa Sampling and analysis of soil*. International co-operative programme on assessment and monitoring of air pollution effects on forests, available at http://www.icp-forests.org/pdf/Chapt_3a_2006%281%29.pdf.

Koch-Steindl H. & Pröhl G. 2001. Considerations on the behaviour of long-lived radionuclides in soil. *Radiat. Environ. Biophys*. 40: 93–104.

Morton L.S., Evans C.V. & Estes G.O. 2002. Natural uranium and thorium distributions in podzolized soils and native blueberry. *J. Environ. Qual*. 31: 155–162.

Mortvedt J.J. 1994. Plant and soil relationships of uranium and thorium decay series radionuclides — a review. *J. Environ. Qual*. 23: 643–650.

Nisbet A.F. & Shaw S. 1994. Summary of a five-year lysimeter study on the time dependent transfer of ^{137}Cs, ^{90}Sr, ^{239,240}Pu and ^{241}Am to crops from three contrasting soil types. 2. Distribution between different plant parts. *J. Environ. Radioact*. 23: 171–187.

Räisänen M.L., Kashulina G. & Bogatyrev I. 1997. Mobility and retention of heavy metals, arsenic and sulphur in podzols at eight locations in northern Finland and Norway and the western half of the Russian Kola Peninsula. *J. Geochem. Explor*. 59: 175–195.

Shahandeh H. & Hossner L.R. 2002. Role of soil properties in phytoaccumulation of uranium. *Water Air Soil Poll*. 141: 165–180.

Sheppard S.C. & Evenden W.G. 1988. Critical compilation and review of plant/soil concentration ratios for uranium, thorium and lead. *J. Environ. Radioact*. 8: 55–285.

Sheppard S.C., Sheppard M.I., Tait J.C. & Sanipelli B.L. 2006. Revision and meta-analysis of selected biosphere parameter values for chlorine, iodine, neptunium, radium, radon and uranium. *J. Environ. Radioact*. 89: 115–137.

Shtangeeva I. 2010. Uptake of uranium and thorium by native and cultivated plants. *J. Environ. Radioact*. 101: 458–463.

Thiry Y., Schmidt P., Van Hees M., Wannijn J., Van Bree P., Ryfyikiri G. & Vandenhove H. 2005. Uranium distribution and cycling in Scots pine (*Pinus sylvestris* L.) growing on a revegetated U-mining heap. *J. Environ. Radioact*. 81: 201–219.

Vandenhove H., Van Hees M., Wannjin J., Wouters K. & Wang L. 2007. Can we predict uranium bioavailability based on soil parameters? Part 2: Soil solution uranium concentration is not a good bioavailability index. *Env. Poll*. 145: 577–586.